

Impacts of REDD+ payments on a coupled human-natural system in Amazonia

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ABSTRACT

We used a hybrid optimization-agent-based model to simulate REDD+ (Reduced Emissions from Deforestation and forest Degradation in combination with conservation, sustainable forest management, and enhancement of forest carbon stocks) payment scenarios to farm households in the old deforestation frontier of Rondônia, Brazil. Payments varied from \$5 to \$30 per ton of net CO₂ either not emitted or removed from the atmosphere relative to a baseline scenario. The impacts of REDD+ were assessed as changes in land use/cover, net CO₂ emissions, program costs, community welfare, and agricultural production. Our results suggest that interventions aimed at zero gross deforestation would require unrealistically large annual disbursements. In contrast, zero net carbon emissions can be achieved at approximately two-thirds the cost with reduced impacts on food production. Overall, simulated payments increased inequality among households, which conflicts with desired REDD+ outcomes. Results suggest that REDD+ might be more successful solely as a climate change mitigation mechanism as opposed to a complex multi-objective development program.

1. Introduction

Recent international negotiations at the Conference of Parties to the United Nations Framework Convention on Climate Change (UNFCCC) recognized REDD+ (Reduced Emissions from Deforestation and forest Degradation in combination with conservation, sustainable forest management, and enhancement of forest carbon stocks) as an official option for climate change mitigation (UN-REDD, 2015; UNFCCC, 2015). Although REDD+ was originally conceived as a performance-based mechanism of payments for environmental services (PES) based on forest CO₂ stocks (West et al., 2018), it has evolved into a multi-objective tool for conservation and development expected to address poverty, biodiversity, land tenure, indigenous rights, and governance issues simultaneously (Angelsen, 2017; UN-REDD, 2015). The challenges of REDD+ implementation has fueled numerous debates and generated criticisms mostly due to potential negative impacts on the

livelihood of forest stakeholders (Caplow et al., 2011; Fletcher et al., 2016).

Appraisals of pilot REDD+ interventions can guide the formulation of better national conservation programs (Groom and Palmer, 2012; Sunderlin et al., 2014; West, 2016), but such assessments are currently limited to short timeframes. The development of REDD+ initiatives that are effective, efficient, and equitable over the long-run rely on scenario-based simulations (Posner et al., 2016; Purnomo et al., 2013). Methodological frameworks based on coupled human-natural systems, like the examples used in this paper, can inform the design and provide insights for ex-ante evaluations of future REDD+ programs (An, 2012; Iwamura et al., 2016; Liu et al., 2007).

Local stakeholder enrollment in REDD+ initiatives can be promoted by PES programs that offset opportunity costs of foregoing conversion of forest to other land uses, assuming that forest owners bear negligible transaction costs (Abram et al., 2016; Hein et al., 2016; Ravikumar

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et al., 2017). A global meta-analysis revealed that the opportunity costs for REDD+ range \$0.05–92.00 US dollars Mg CO_2^{-1} , with average and median values of \$7 and \$3 Mg CO_2^{-1} , respectively (Dang Phan et al., 2014). The highest values were for Southeast Asian forests threatened by oil palm plantations. The mean opportunity cost in Latin America was \$5.4 Mg CO_2^{-1} , with a range of \$0.2–13 Mg CO_2^{-1} in the Brazilian Amazon (Börner and Wunder, 2008). Based on household survey data, Ickowitz et al. (2017) reported substantially higher opportunity costs to smallholders over a 30-year time horizon with a 9% discount rate, ranging \$11–77 Mg CO_2^{-1} across five sites in the same biome. In contrast, Nepstad et al. (2007), simulating returns to soy and cattle production, estimated an opportunity cost of only \$1.5 Mg CO_2^{-1} across Amazonian Brazil. In general, REDD+ opportunity costs vary with forest carbon stocks, the profitability of alternative land uses, discount rates, and time horizons (e.g., Cacho et al., 2014; Irawan and Tacconi, 2009; Lu and Liu, 2013). Thus, evaluation of the potential impacts of REDD+ programs need to be site- and agent-specific (Balderas Torres et al., 2013), and consider income variations (Ickowitz et al., 2017).

An additional challenge for any given REDD+ intervention is that conservation efforts tend to be static whereas opportunity costs and other factors that drive decision-making are dynamic (Liu et al., 2007; Parker et al., 2003). To persist, these interventions need to be designed in ways that allow adjustment of compensation levels as economic conditions change (Cacho et al., 2014). The overall dynamism of decision-making also suggests that static analyses will fail to capture the complexity and nonlinearity of coupled human-natural systems (Kelley and Evans, 2011; Matthews et al., 2007). Coupled human-natural systems represented in simulation models, e.g., agent-based models (ABM), can provide an appropriate framework to explore these dynamic conditions (An, 2012; Iwamura et al., 2016). In this paper we used the *SimREDD+* hybrid optimization-ABM model (West et al., 2018) to evaluate the potential impacts of REDD+ payment scenarios on land-use/cover change (LUCC), net CO_2 emissions, program costs (defined as the sum of all REDD+ payments to households), and community welfare (defined as the sum of all farm-based annual household profits) within settlements in the Ouro Preto do Oeste region, an old deforestation frontier in Rondônia, Brazil.

2. Materials and methods

2.1. Study region

Settlements in the study region cover approximately 5120 km^2 of Rondônia State, Brazil (Fig. 1), including the municipality of Ouro Preto do Oeste and its five contiguous neighbors, Vale do Paraíso, Nova União, Teixeiraópolis, Urupá, and Mirante da Serra, all of which are near federal highway BR-364. The area lies within the “arc of deforestation” in the Brazilian Amazon and it is a well-documented heavily-deforested old frontier settled in the 1960–1980s, largely in response to government-sponsored development programs (Caviglia-Harris, 2004; Roberts et al., 2002; Sills and Caviglia-Harris, 2009).

The majority of households in the region (98%) are small-scale agricultural producers with an average property size of 65 ha (range: 2–240 ha), while the average size for the remaining medium and large lots is 745 ha (range: 240–3000 ha; West et al., 2018). In general, households in our landscape are considered poor by national and international standards (Caviglia-Harris, 2005). Most farm households initially focused on annual and perennial crops when migrating to the region in the 1970s. By 1996 annual and perennial crops fell to 45% of the total agricultural income, with dairy largely occupying the remaining 55%. By 2009 dairy production remained at a similar percentage (53%) of total agricultural income, while beef production increased to 32% and annual and perennial crop income fell to 15%. In sum, cattle activities focused on dairy and beef production increased from 55% of income in 1996 to 85% of total agricultural income by

2009 (Caviglia-Harris et al., 2015).

2.2. Model description

SimREDD+ adopts a bottom-up approach that captures the decision-making processes of local farmers (West et al., 2018). The model was parameterized and validated with socioeconomic survey data from households in the study region in 2009 (Caviglia-Harris et al., 2009). Its spatial component is composed of farm boundaries (Fig. 1), a land-use/cover classification map from 1996 (initial year of the simulations) that indicates mature forest, secondary forest, and agricultural areas (Fig. S1; Toomey et al., 2013), and additional maps that represent areas most suitable for deforestation or agricultural abandonment. Additionally, the carbon-density map developed by Baccini et al. (2012) and the carbon stock estimates for the vegetation replacing Amazonian forests by Fearnside (1996) were used to estimate net CO_2 emissions from LUCC processes (Figs. S2 and S3). Model simulations are evaluated in terms of the “3E” framework (efficiency, effectiveness, and equity) proposed for assessment of REDD+ initiatives (Angelsen, 2008; Vatn and Vedeld, 2013).

The model, implemented in NetLogo v.5.3.1 (Wilensky, 1999), contains two types of agents: farm households (or farmer-agents) and land-use/cover patches. Farmer-agents seek to maximize their farm-based profits by choosing the optimal allocation of land for agriculture given a REDD+ scenario. While most farmers in our study region focused on dairy and beef production, other households emphasized annual and perennial crop production. We converted all agricultural production to rice-equivalent units based on their energy values in order to incorporate the mixture of crops into the model. This procedure was necessary to reach an analytical solution to the optimization problem in *SimREDD+* (Barnum and Squire, 1979; West et al., 2018). Land-use/cover raster cells in the model respond to individual farmer-agent landowner decisions. LUCC is tracked during the simulations and net CO_2 emissions are calculated at model equilibrium. Simulations start based on the 1996 landscape, when there was still a considerable amount of mature forests in the region and international negotiations about the inclusion of avoided deforestation activities in a future climate treaty were underway (Tolba and Rummel-Bulska, 1996).

Except for a few large land-holdings that are excluded from our analysis, farms cover an area of 513 thousand ha of which, in 1996, 272 thousand ha were under agricultural use (53%), 184 thousand ha were mature forest (36%), and 56 ha were covered by secondary forest regrowth (11%). Payments ranged \$5–30 per ton of net CO_2 not emitted or removed from the atmosphere. Given a simulated scenario, the farmer-agent chooses the optimal allocation of land to agriculture and forest within his property and defines the LUCC necessary to achieve the optimal configuration at model equilibrium. Although 1996 was selected as the initial year of the simulations, land use/cover at equilibria are the same irrespective of the initial land-use/cover configuration. Given the assumptions that areas set aside for secondary forest development eventually attain the carbon stocks of mature forest (Poorter et al., 2016), choosing different start years for the simulations implies changes only in the initial proportions of mature and secondary forests. Thus, only the net carbon emissions from LUCC are affected if the initial year of the model is changed (West et al., 2018).

2.3. REDD+ payments scenarios

REDD+ payments are integrated into the profit maximization decisions of the farmer-agents. PES are based on the net carbon emissions from avoided conversion of forest to agricultural land and net carbon sequestration from forest recovery on abandoned agricultural patches (\$ Mg CO_2^{-1}), in accordance to the rules set by voluntary carbon market standards (Verified Carbon Standard, 2017). The REDD+ payment scenarios simulated were assessed in terms of (1) changes in the areas of mature forest, secondary forest, and agricultural land, (2) total

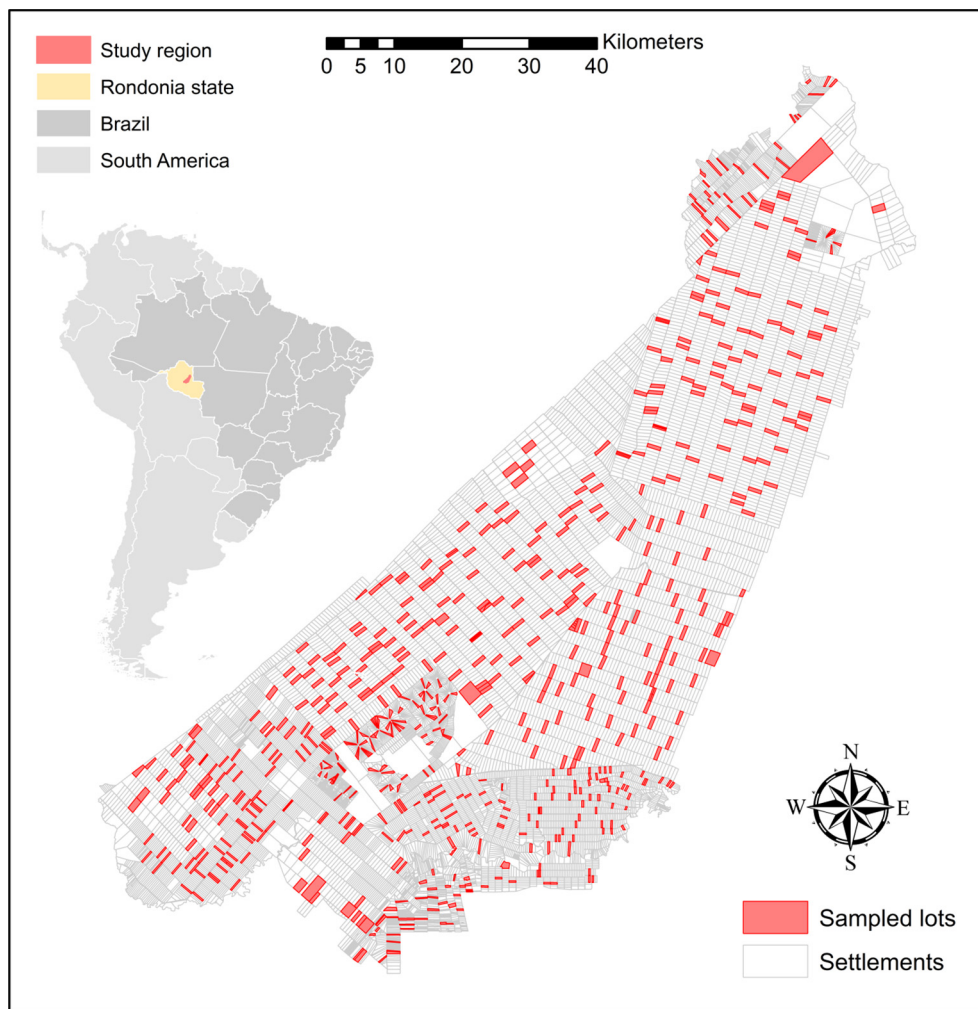


Fig. 1. Study region in Rondônia State, Brazil.

net CO₂ emissions or uptake, (3) changes in community welfare (defined as annual household profit from farm-related activities), (4) welfare disparities between household economic classes (e.g., wealthy versus poor), and (5) annual program costs at model equilibrium. Farm household income is proportional to property size and was used to classify households into four economic groups based on quartile thresholds. Simulated annual REDD+ payments ranged \$5–30 Mg CO₂^{−1} to reflect historical and current prices of carbon offsets based on market transactions reports (Hamrick and Goldstein, 2016).

2.4. Profit maximization calculations

Households in SimREDD+ behave as profit maximizers under perfect labor market conditions (Angelsen, 1999). The LUCC decision-making process is based on dynamic maximization of a farm profit function (Eq. (1)) adapted to include the financial benefits of REDD+:

$$\Pi = p_A \alpha A_t^\beta L_t^\varphi + p_E (F - A_t) - w(L_t + \gamma D_t) - C_A A_t - C_D D_t \quad (1)$$

where p_A is the price of the agricultural production output; the term $\alpha A_t^\beta L_t^\varphi$ represents a Cobb-Douglas production function based on agricultural area (A_t) and agricultural labor (L_t) at time t ; where β and φ are the land area and labor output elasticities, respectively, and α is the total factor productivity; w is the wage rate associated with agricultural labor, deforestation, and site preparation labor (γ) per unit of deforested area (D_t); C_A and C_D are non-labor costs associated with maintaining agricultural areas (A_t) and deforestation and site preparation

costs required per hectare of deforested area (D_t), respectively. A Cobb-Douglas function was chosen because of its closed-form solution property and its flexibility, given that the use of inflexible functional forms may result in misleading predictions and policy recommendations (Angelsen, 1999; Bronfenbrenner and Douglas, 1939). The profit function includes the financial benefit from PES with the term $p_E (F - A_t)$, where p_E is the REDD+ payment (\$ CO₂^{−1}) based on the total farm area (F) subtracted from the current agricultural land use (A_t). This formulation also implies that reforestation is represented by a negative input of A_t (decrease in the agricultural land).

LUCC decisions in SimREDD+ are framed as an optimal control problem (Michel, 1982), where farmer-agents optimize farm profits by reaching the optimal level of agricultural land (and forests) in the property at steady state under a given PES scenario and inequality constraints. Eq. (2) represents the analytical solution to the maximization exercise (see West et al., 2018, for the step-by-step derivation of the optimal control problem):

$$A^* = \left(\frac{w \beta^{1-\frac{1}{\varphi}} \alpha^{-\frac{1}{\varphi}} p_A^{-\frac{1}{\varphi}} (r C_D + p_E \delta + C_A + r \psi \gamma + r \psi + \psi)^{\frac{1}{\varphi}-1}}{\varphi} \right)^{\frac{\varphi}{\beta + \varphi - 1}} \quad (2)$$

where A^* is the optimal agricultural land at equilibrium and ψ is the shadow value associated with the maximum deforestation constraint (i.e., farm size). Finally, the optimization submodel calculates community welfare across household socioeconomic groups, based on the sum of all individual annual household farm-based profits at

equilibrium, as well as the total annual REDD+ PES disbursements given the household enrollment rate in the program (see West et al., 2018, for details).

2.5. LUCC allocation submodel

Once the optimal agricultural land area is defined, farmer-agents adjust land use/cover in their lots to achieve that level. When deforestation is required, farmer-agents first clear their secondary forests (see Caviglia-Harris et al., 2015), with preference to patches with higher deforestation suitability, which was calculated with a machine-learning algorithm based on artificial neural networks and 23 spatially-explicit variables described in West et al. (2018). If the optimal amount of agricultural land is not attained by the clearance of all secondary forests, farmer-agents convert mature forest patches. Deforestation is then allocated to the patches with the highest suitability levels until the optimal agricultural area is reached or the entire farm is under agricultural use. When the optimal amount of agriculture land is less than the amount previously used for agriculture, farmer-agents abandon agricultural patches with preference to those with higher abandonment suitability (i.e., closer to forest areas), until the optimal level is reached. Abandoned patches are naturally regenerated and reach the status of mature forests at equilibrium. Finally, the size of each land-use/cover class and the net carbon balance from the LUCC (CO₂ emissions minus sequestration) are reported at the equilibrium (see West et al., 2018, for details).

3. Results

3.1. Land-use/cover change

REDD+ payments based on Mg CO₂ had direct impacts on land-use/cover configurations at model equilibria (Figs. 2 and 3). Had REDD+ initiatives been implemented by 1996 and projected to equilibrium, payments of about \$30 Mg CO₂⁻¹ would have been required to preserve virtually all (97%) of the 184 thousand ha of mature forests present at the beginning of the simulation. That scenario would have been accompanied by a substantial decrease (−72%) in the 272 thousand ha of agricultural land in 1996 relative to the baseline equilibrium without REDD+. For payments below \$10 Mg CO₂⁻¹, no natural regeneration of secondary forests occurred. With payments of \$15 Mg CO₂⁻¹, 70% of the remaining mature forest in 1996 would be preserved (128 thousand ha), along with an expansion of the proportion of secondary forest regrowth in the landscape from 11% to 27% (56 to 138 thousand ha), and a decrease of the proportion of agricultural land from 53% to 48% (272 to 246 thousand ha). Simulated payments of \$5, \$10, \$15, \$20, \$25, and \$30 Mg CO₂⁻¹ yielded avoided deforestation of 44, 80, 128, 147, 153, and 179 thousand ha of mature forest, respectively, and allowed regeneration of 10, 55, 138, 177, 206, and 258 thousand ha of secondary forests, respectively, when compared to the baseline scenario without REDD+.

3.2. Net CO₂ emissions

In the absence of REDD+, total net carbon emissions due to LUCC from 1996 until the model's baseline equilibrium was reached were estimated at 98 Tg CO₂. Zero net emissions were achieved with payments of \$10–\$15 Mg CO₂⁻¹, mostly due to forest regeneration in previously established agricultural areas (Fig. 3). Consequently, in all scenarios with REDD+ PES greater than \$15 Mg CO₂⁻¹, the landscape became a carbon sink at equilibrium. Carbon uptake estimates from secondary forest regrowth ranged 11–84 Tg CO₂ for payments of \$15 and \$30 Mg CO₂⁻¹, respectively. PES of \$5, \$10, \$15, \$20, \$25 and \$30 Mg CO₂⁻¹ yielded net carbon emission reductions of 22, 52, 109, 134, 149, and 182 Tg CO₂, respectively, when compared to the baseline scenario without REDD+ at the landscape level.

3.3. Annual program costs

A positive nonlinear relationship was observed between the level of REDD+ payments offered to households and annual REDD+ costs due to household enrollment in the conservation program (Fig. 3). A program that offers payments of \$5 Mg CO₂⁻¹ would be associated with PES disbursements of \$5.8 million year⁻¹, while payments of \$10 and \$30 Mg CO₂⁻¹ would require investments of \$29 and \$283 million year⁻¹, respectively. Such nonlinear relationships derive from the capacity of the REDD+ PES to offset household opportunity costs associated with agricultural activities. Agricultural costs increase linearly with labor and area, while agricultural outputs decrease relative to scale (West et al., 2018). As per unit PES increase, REDD+ PES expenditures increase linearly only if the area allocated to forest remains the same. However, increasing PES encourages more land to be allocated for forest protection or recovery, which causes REDD+ expenditures to rise faster than the payments offered. Other estimated annual costs associated with PES of \$15, \$20, and \$25 Mg CO₂⁻¹ scenarios were \$87, \$141, and \$194 million, respectively. Our analyses did not include management or transaction costs associated with REDD+ (Rendón Thompson et al., 2013).

3.4. Community welfare

Welfare in the settlements increased substantially in the simulated REDD+ scenarios. Payments of \$5, 10, 15, 20, 25, and 30 Mg CO₂⁻¹ increased community welfare by 8%, 36%, 102%, 164%, 227%, and 335%, respectively (Fig. 3), but the largest and wealthiest households captured most of those benefits. For example, PES of \$5 Mg CO₂⁻¹ did not affect the welfare of the lower two quartiles of the population based on household income (poorest farmers), but welfare increased by 3% and 25% for the third and fourth quartiles, respectively (Table S1). Increases in inequalities simulated by SimREDD+ were driven by decreasing returns from agricultural outputs that result from increasing inputs (i.e., increases in agricultural land and labor lead to less than proportional increases in outputs). The opportunity cost of converting land at the margin to agriculture decreases with property size. When enrollment in REDD+ programs is voluntary (Wunder, 2015), households would simply opt out of the program if they were to suffer welfare reductions due to their enrollment. Given that costs are held constant, welfare, as represented by annual farm household profit, can only increase or remain the same under any simulated REDD+ scenario. Still, the SimREDD+ model does not capture changes in livelihood costs associated with welfare improvements, which can be expected to rise as wealth increases in our study region.

3.5. Agricultural production

Agricultural production declines with REDD+ payments because they stimulate reductions in the proportion of land dedicated to farming in the optimal land-use/cover bundle that maximizes household profit (Fig. 3). For the \$5 Mg CO₂⁻¹ PES scenario, agricultural production decreased by −4% compared to the baseline equilibrium but then continued to decrease nonlinearly by −13%, −30%, −41%, −48%, and −61% when REDD+ payments increased to \$10, \$15, \$20, \$25, and \$30 Mg CO₂⁻¹, respectively. As with the changes observed in welfare, the wealthiest households (fourth quartile) experienced the largest decreases in agricultural production at the margins. For example, REDD+ payments of \$15 Mg CO₂⁻¹ did not change the production of the poorest group, but production by the wealthiest declined by −26%. Under the most extreme payment scenario (\$30 Mg CO₂⁻¹), the same groups produced −34% and −74%, respectively. Shifts in food production were a direct consequence of changes in the optimal bundle of forest and agricultural areas within the farms at model equilibria. Hence, as with welfare, the results for agricultural production were driven by the decreasing returns to scale of the production

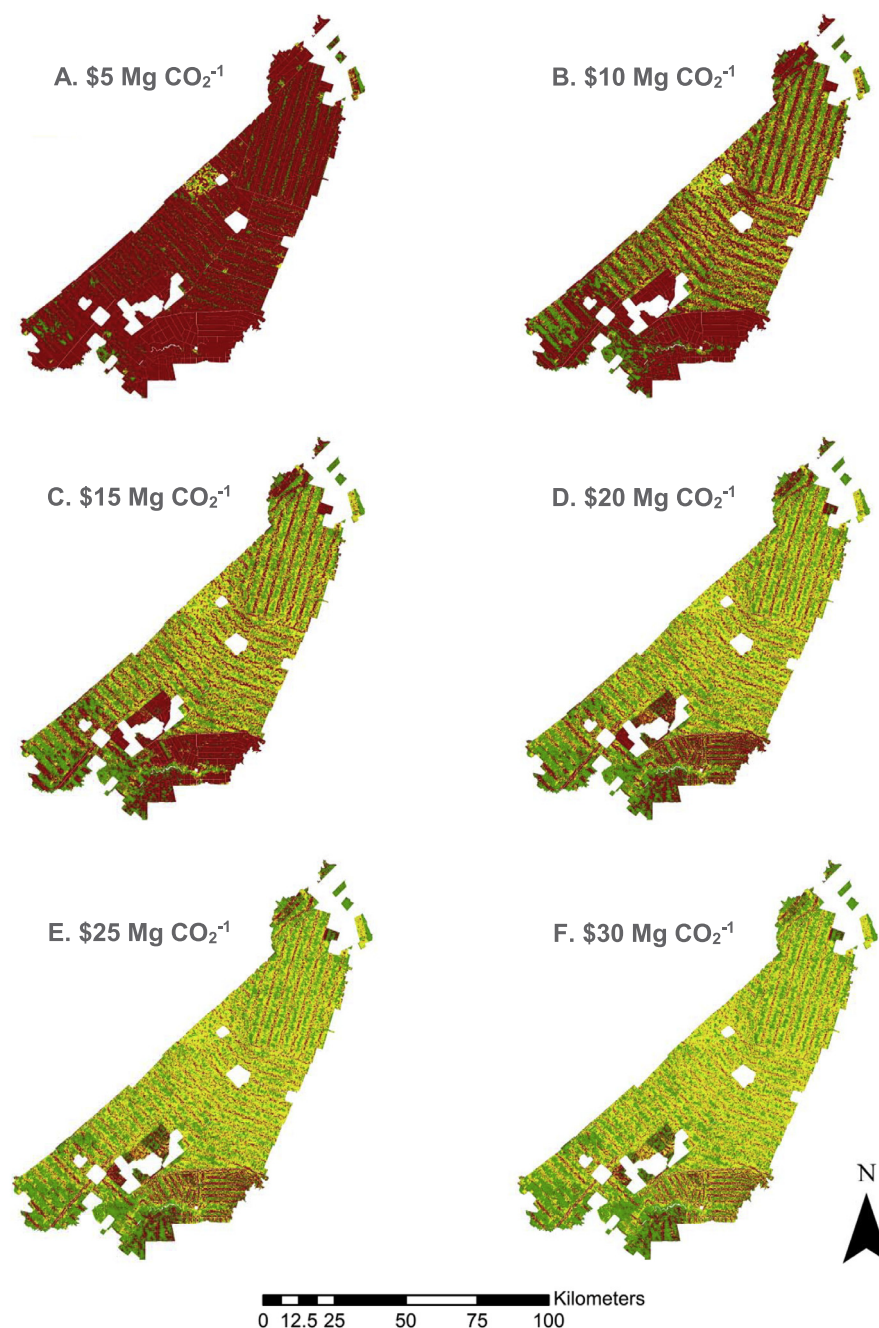


Fig. 2. Land-use/cover changes from 1996 to model equilibria for different per-Mg CO₂ REDD+ payments. Green, yellow, and red patches represent mature forests, secondary forests, and agriculture, respectively. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

function in SimREDD+.

4. Discussion

REDD+ is portrayed as a cost-effective mechanism to mitigate climate change (Börner et al., 2010; Groom and Palmer, 2012; Lubowski and Rose, 2013). Substantial sums have been made available to support REDD+ initiatives in the tropics. These include billion-dollar bilateral agreements between donors and developing countries from the World Bank, UN-REDD, and the Norwegian government (Angelsen, 2017; Fearnside, 2012; Fletcher et al., 2016). Our simulations indicate that REDD+ payments of \$30 Mg CO₂⁻¹, associated with annual program disbursements of \$238 million, could have avoided the clearing of virtually all mature forests that remained in our study region.

Nevertheless, it is hard to conceive that annual REDD+ PES of hundreds of millions of dollars to a relatively small area on an Amazonian deforestation frontier could be sustained. Furthermore, since the simulations assume idealized REDD+ circumstances (e.g., they do not account for implementation, regulation, monitoring, and other administrative costs), the annual expenditures on PES reported here do not capture the total cost of REDD+ programs (Luttrell et al., 2018). Still, the costs associated with our simulated scenarios are in agreement with other studies in the literature. Kindermann et al. (2008) estimated the cost of reducing tropical deforestation by 50% for 2005–2030 at \$17–\$28 billion year⁻¹. These estimates contrast with the estimated \$5 billion year⁻¹ costs of forest protection in eight countries responsible for 70% of emissions from deforestation presented in the highly-cited Stern Review (Pedroni et al., 2009), as well as with the approximate

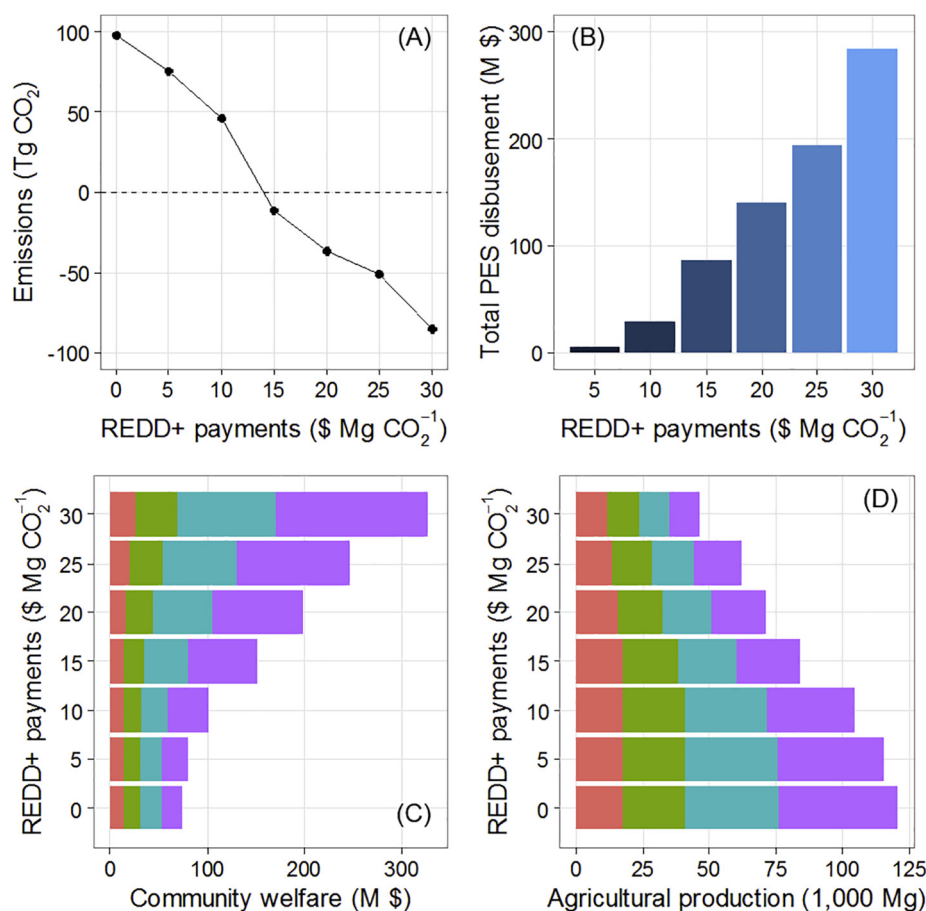


Fig. 3. Impacts of REDD+ payment scenarios based on carbon stocks on the coupled human-natural system. Panel A represents net CO₂ emissions from 1996 to model equilibria. Panel B is the total annual disbursements of REDD+ on payments for environmental services (PES) to households. Panels C and D display community welfare based on annual farm household profit and annual agricultural production based on rice-equivalents, respectively. In the latter panels, red, green, blue, and purple segments represent the poorest to the wealthier household groups by quartiles, respectively.

\$45 million over a 10-year period estimated by Börner et al. (2010) for settlements in Amazonian Brazil based on static opportunity cost calculations. Comparatively, the cost of Brazil's strategy to reduce deforestation to 80% of the historical rate, presented in 2009 at the Conference of Parties (COP15) to the UNFCCC, was estimated at approximately \$6–18 billion (Nepstad et al., 2009). Nonetheless, the decisions incorporated into the Paris Agreement might create a brighter future for REDD+, which could partially benefit from the \$100 billion annual investment to be provided by developed countries for climate change mitigation and adaptation by 2020 (UNFCCC, 2015).

Brazil pledged to achieve zero *illegal* deforestation and restore 12 million hectares of forests by 2030 as part of its national contribution towards achieving the objective of the UNFCCC (Brasil, 2015). However, Brazil's earlier goal was to achieve zero *net* greenhouse gas emissions from LUCC by 2015 (Brasil, 2008). Had the nation maintained that initial goal, SimREDD+ simulations suggest that REDD+ PES of close to \$15 Mg CO₂⁻¹ would be required to achieve zero net carbon emissions from the settlements in our study region. This would imply annual REDD+ PES disbursements of approximately \$87 million to preserve an area of 5120 km². Although still substantial, these estimated costs are far less than the amount required for conservation of all mature forests in our old deforestation frontier. REDD+ disbursements vary with the number of households enrolled in the PES program, while enrollment is driven by whether PES offset the opportunity costs of other land uses. Most households in our study region run small farms for milk production, the main driver of local income increases between 1996 and 2009 (Caviglia-Harris, 2004; Sills and Caviglia-Harris, 2009). Other settlement regions in the Amazon differ from ours, which means the impacts of REDD+ PES would also likely differ. For example, in areas where large and very productive farms dominate, the costs of avoided deforestation or promoting secondary forest recovery are likely

to be even higher than in our study region (Dang Phan et al., 2014).

We did not impose deforestation constraints based on environmental regulations on the farmer-agents in our simulations because it is not clear that those regulations are effective in our study region. The Brazilian Forest Code sets specific thresholds for forest conservation on rural properties in the Amazon (i.e., *Legal Reserves* and *Permanent Preservation Areas*). Whereas a series of recent policy instruments reportedly curbed deforestation in the country (Nepstad et al., 2014), they have arguably not yet affected the widespread non-conformance with environmental regulations by small farmers (Nunes et al., 2015). Moreover, our model assumption is corroborated by Godar et al. (2014) who found that deforestation on the largest properties (≥ 2500 ha) had declined by 63% from 2005 to 2011 but increased by 69% on the smallest properties during the same period.

In our landscape, PES of \$5–30 Mg CO₂⁻¹ were equivalent to area-based payments of \$100–650 ha⁻¹ year⁻¹ due to high carbon stocks in the native vegetation compared to agricultural lands (Baccini et al., 2012; Fearnside, 1996). However, in areas with low forest biomass but high biomass stocks in the baseline scenario, such as where heavily degraded forests are converted to palm oil plantations (Gaveau et al., 2016), payments per ton of CO₂ could be equivalent to much lower per-area PES. It is important to note that annual per-hectare payments of \$100–650 are much higher than those recorded to date. For example, Arriagada et al. (2012) reported annual payments of approximately \$43–62 ha⁻¹ in Costa Rica, (but estimated the real costs of increasing forest conservation, i.e., additional when compared to the baseline, at \$255–\$382 ha⁻¹). Similarly, the Socio Bosque program in Ecuador offered annual payments of \$30 ha⁻¹ for the first 50 ha enrolled, \$20 ha⁻¹ for additional 50–100 ha, with continuously decreasing payments until reaching \$0.5 ha⁻¹ for areas $> 10,000$ ha (de Koning et al., 2011). In Mexico, conservation payments ranged from \$22 ha⁻¹ year⁻¹

in the federal program to \$111 ha⁻¹ year⁻¹ in the Probosque state program (Balderas Torres et al., 2013), whereas PES from other programs ranged \$8–12 ha⁻¹ year⁻¹ (Honey-Rosés et al., 2009). Lastly, the Peruvian National Forest Conservation Program for Climate Change Mitigation provided PES incentives of approximate \$3.5 ha⁻¹ year⁻¹ of forest enrolled in the program by local communities (Börner et al., 2016). If similar payment ranges were adopted by REDD+ or other conservation programs in Brazil, our simulations suggest that meaningful climate change mitigation results would not be achieved in our study region.

Increased inequalities among household economic groups were driven by the decreasing returns to scale behavior of the production function and shifts in agricultural production due to the new LUC decisions (West et al., 2018). Our model behavior is consistent with the findings of several econometric studies focused on small farm household production (Barrett et al., 2010; Carletto et al., 2013; Graeub et al., 2016; Larson et al., 2014; Neumann et al., 2010). As REDD+ payments increase, the optimal land-use/cover bundle between agricultural land and forests change, but the magnitude of the shifts varies with farm size. Overall, for the same PES level, profits of large landowners are maximized with a higher proportion of land in forests than for owners of smaller tracts. Assuming no misspecification of production functions (Griliches and Mairesse, 1995; West et al., 2018), decreasing returns to scale implies that equity of REDD+ payments will never be achieved in certain communities. In our settlement region, PES will always increase inequalities between poorer and wealthier households as long as the payments are proportional to committed areas or carbon stocks and REDD+ participation is voluntary. Most importantly, increases in overall community welfare from REDD+ benefited the wealthiest households the most. Equality is intrinsic in the scenarios simulated in SimREDD+ because PES and LUC decisions are based on individual and marginal opportunity costs. Yet, there is no equity associated with the distribution of program benefits because they are mostly captured by the wealthy. Since increases in welfare are often linked to increases in livelihood costs (Douglas et al., 2014), REDD+ might cause undesired economic impacts on the poorest groups.

Other researchers noted the equity pitfalls associated with REDD+ and PES (Andersson et al., 2018). Pascual et al. (2014) highlight that efficient PES schemes might be designed based on household opportunity costs, but their efficiency gains could be short-lived if the intervention is considered inequitable by program participants, who would then quit. To avoid this problem, Börner et al. (2016) suggested the use of fixed household payments. Inequitable outcomes conflict with the “3E” framework (i.e., Effectiveness, Efficiency, and Equity) often advocated for REDD+ interventions (Angelsen, 2008) and underscores concerns about the scope and purpose of REDD+. Could the shift of REDD+ from a solely PES-based initiative into a multi-objective concept (Angelsen, 2017; Pinho et al., 2014) be detrimental to climate change mitigation? A payment mechanism designed to maintain and enhance carbon stocks in natural systems is clearly a PES that could be integrated into national conservation and development plans, but could also exist as a strict-PES scheme alone (Ferraro, 2011; Pagiola et al., 2005; Wunder, 2015). This idea is aligned with Coasean policy approaches (Diswandi, 2017), emphasizing the importance of aggregate gains or losses by different economic agents but not their equity implications (Pascual et al., 2010). Furthermore, it is well recognized that opportunity costs can vary greatly among LUC agents, from subsistence farmers to large-scale agricultural companies (Dang Phan et al., 2014; Ickowitz et al., 2017). This issue creates a loophole that will be unlikely to help poor farmers increase their welfare if compensation is based on a farmer's estimated revenue (Pirard, 2008), an approach that might never be observed in the real world. Our simulations suggest that REDD+ payments represent a more effective tool for climate change mitigation if equity issues at the regional level are either overlooked or otherwise addressed. Although an alternative design for the PES could lessen the discrepancy of REDD+ benefits between wealthier and

poorer farmers in our study region (e.g., de Koning et al., 2011), it could also limit land enrollment in the conservation program if its capacity to offset opportunity costs is not optimized. As argued by Ickowitz et al. (2017), if households are offered a flat PES rate equivalent to the local average opportunity cost, REDD+ would have a “pro-poor” effect on low-income farmers who would therefore be more inclined to participate in the program than wealthier householders. However, what would happen if poorer farmers become wealthier because of the REDD+ payments and their opportunity costs increase above the flat payment rate? In this regard, the dynamic LUC optimization framework in SimREDD+ is a suitable tool to evaluate potential long-term outcomes from REDD+ scenarios by addressing the limitations of static opportunity cost calculations (Börner et al., 2010; Ickowitz et al., 2017; West et al., 2018).

Our simulations also provide some insights into the effect of PES on food security, which is of particular relevance for REDD+ insofar as smallholder agriculture is the backbone of food security in developing countries (Tscharntke et al., 2012). Although perspectives contrast greatly, it is clear that pressures on the global food system will increase drastically due to increasing populations and per-capita consumption, growing competition for inputs, and climate change effects (Godfray and Garnett, 2014). In the scenarios simulated in this study, regional agricultural production decreased substantially with the highest REDD+ payments, although some food production persisted at model equilibria. Reductions in agricultural production are of concern not only for food security but also because of the substantial risk of “leakage” (i.e., displacement as opposed to reduction of carbon emissions; Aukland et al., 2003). Estimating leakage is beyond the scope of this study but a few considerations are noteworthy. Under the nested governance approach advocated for national REDD+ programs (Pedroni et al., 2009; West, 2016), and arguably endorsed by the UN-REDD initiative (UN-REDD, 2011; UN-REDD, 2015), leakage from subnational REDD+ activities to different locations within the same country are identified through national monitoring (e.g., UNFCCC decisions 4/CP.15, 1/CP.16, and 11/CP.19). In contrast, concerns about national sovereignty limit consideration of international leakage, which therefore needs to be avoided with the implementation of specific leakage mitigation measures.

The assumption that households maximize profits was made in full recognition that economic optimization may not motivate many LUC decision-makers (Jager et al., 2000). For example, in a study of the potential outcomes from REDD+ payments to a forest community in Honduras, Plumb et al. (2012) reported that not all LUC behavior was economically driven and that not all land use values are easily calculable in monetary terms, which in turn can lead to underestimation of opportunity costs. Furthermore, as noted by Iwamura et al. (2016), simulation models are simplifications of reality and their results should be interpreted with caution. The contribution of this work is more important in terms of understanding nonlinear feedbacks in our coupled human-natural system related to LUC, CO₂ emissions, food production, and welfare resulted from direct REDD+ payments to small farm households, as opposed to particular output values. Nevertheless, when applied and interpreted with caution, simulation models of coupled human-natural systems can help in the assessment of the effectiveness, efficiency, and equity of conservation and development interventions (An, 2012; Angelsen, 2008; Iwamura et al., 2016).

5. Conclusions

We used SimREDD+, a coupled human-natural system model, to explore the potential impacts of REDD+ payments on LUC, net carbon emissions, welfare, and agricultural production on an old deforestation frontier in the Brazilian Amazon. Simulation results suggest that REDD+ programs aimed to achieve zero deforestation may require huge annual payments. In contrast, if the intervention goals shifted to achieving zero net carbon emissions, REDD+ disbursements decrease

by approximately two-thirds. Furthermore, decreases in agricultural production, and consequently REDD+ impacts on food security, would not be as extreme.

The simulated scenarios revealed that REDD+ payments would increase inequality among households in our study region, assuming decreasing returns to scale and the existing distribution of land ownership. This would conflict with the “3E” framework goals advocated for REDD+. Simulations suggest that there is a tradeoff between the effectiveness of REDD+ as a climate change mitigation activity and equity. While there is no question that social problems must be confronted by appropriate and preferably synergistic development agendas, results from this study suggest that REDD+ might be more successful as a PES scheme focused solely on climate change mitigation. Future research shall investigate the effects of alternative PES designs to mitigate socioeconomic inequalities resultant from REDD+ activities.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <https://doi.org/10.1016/j.ecoser.2018.08.008>.

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